TIE LAKE
PHOSPHORUS BUDGET AND RELATED WATER QUALITY IN 1980 - 1981

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EXECUTIVE SUMMARY

Tie Lake is a shallow, hard water lake located about 35 km. east of Cranbrook. It has a surface area of 126 hectares and is comprised of three basins arranged in an S-shape. The water residence time is fairly long for a small lake at approximately 3 years with a large portion of the runoff entering the lake via groundwater. There are approximately one hundred lakeside dwellings and the lake receives fairly high recreational use in the summer particularly water skiing.

In the late summer of 1980 a blue-green algal bloom developed turning the lake a turbid grass-green colour and producing an unpleasant fishy odour. Many of the residents registered their concern with the Waste Management Branch in Cranbrook and eventually formed a Ratepayer's Association to deal with water quality problems. The Waste Management Branch subsequently undertook a study to determine the reasons for this accelerated eutrophication.

A study had been conducted by the Water Investigations Branch in 1972 and 1973 which concluded that the lake was mesotrophic and phosphorus limited. The study recommended that lakeside septic systems be located a minimum of 33 meters from the lakeshore or any inflowing streams. The study suggested that failure to adhere to this set-back coupled with increased residential development would lead to the deterioration of lake water quality.

The 1980-81 study found that the lake phosphorus concentration was two to three times that which was recorded in 1972-73. The major sources of this phosphorus in roughly equal proportions were determined to be lakeside septic systems and livestock ranging close to the lake. These elevated phosphorus levels gave rise to the algal bloom in 1980 which persisted until May 1981. At this time high levels of dissolved calcium in conjunction with photosynthetic consumption of carbon dioxide initiated a precipitation of calcium carbonate which in turn coprecipitated about half of the phosphorus in the lake. This calcite precipitation also resulted in the removal of a large portion of the algal bloom and a clearing of the lake in late May 1981.

With present or increased levels of phosphorus input the lake will eventually suffer another undesirable algal bloom similar to that of 1980. As suggested by the Water Investigations Branch study, calcium carbonate/phosphorus coprecipitation may periodically lower lake phosphorus concentration delaying its increase to problem levels.

The depletion of oxygen under ice, which causes periodic fish kills, was determined to be primarily a function of the large sediment surface area/lake volume ratio (low mean depth). Eutrophication of the lake will increase the oxygen depletion rate over a number of years making winter fish kills a more frequent occurence.

Several measures which could be implemented to reduce phosphorus b ldup in the lake were presented. A summary of those measures is as follows:

1. Curtail the use of the Tie Lake seeding pasture. This would eliminate an estimated 30% of the total annual phosphorus loading and result in a similar reduction in lake phosphorus concentration. The pasture could be relocated out of the drainage basin.

2. Divert a flow of approximately 1.0 cfs from Little Sand Creek thereby increasing the flushing rate of the lake and decreasing the lake phosphorus concentration.

3. Apply a quantity of quick lime to the lake to increase pH and dissolved calcium levels thus initiating calcite precipitation sooner than would naturally occur. The addition of lime should be done in conjunction with spring water quality sampling to reduce phosphorus levels and present a problem algal bloom from occurring. This of course is a treatment rather than a remedy and would have to be done on an ongoing basis if the phosphorus loading is not reduced.

4. Improve operation and upgrade individual sewage disposal systems. This would result in some reduction of phosphorus loading to the lake.

5. Remove individual sewage disposal systems from the basin. This is possible through construction of a collection system and treatment plant with discharge to infiltration lagoons outside the drainage basin. The use of holding tanks and removal of sewage to another treatment plant or disposal facility is another way of achieving this result.

Without the benefit of detailed cost/benefit analysis, alternatives 1, 2 and 3 were identified as being those which should receive most active consideration.

1. INTRODUCTION

Tie Lake is located 35 km east of Cranbrook at latitude 49° 25' N and longitude 114° 19' W. The lake lies at an elevation of 849 meters while 3 km north the Lizard Range of the Rocky Mountains rise to elevations of 2900 meters. The soils of the area are fluvio-glacial or morainal deposited by glaciers and thus modified into kames, eskers and generally irregular topography. The vegetation of the area is dominated by interior Douglas fir with some Lodgepole and Ponderosa pine and Western Larch. Birch and Poplar are found in low lying wet seepage areas and around the lake perimeter.

The lake itself is shallow, with a mean depth of about 2 meters and a maximum depth of about 5 meters, and covers an area of 126 hectares. The drainage basin is poorly defined covering an estimated area of about 780 hectares. The lake drains into Little Sand Creek which flows southeast along the foot of the Lizard Range and intercepts mountain runoff from reaching the lake. Little Sand Creek joins Sand Creek and flows into Lake Koocanusa about 12 km south of Tie Lake.

The lake has periodically suffered from excessive algal blooms which prompted a study by the Water Investigations Branch in 1972-73 (Buchanan & Goddard,1976). In the late summer of 1980 the worst algal bloom in memory occurred raising the concerns of lakeside residents who subsequently formed a Ratepayer's Association to deal with water quality problems. The Waste Management Branch - Kootenay Region began a relatively detailed study of the lake in November 1980 which lasted through to September of 1981. The purpose of the study was to

quantify the lake's productivity and to develop a nutrient budget to determine if eutrophication was natural or the result of human activity.

This report presents the findings of the study along with predictions for future impacts and some suggested methods for reducing eutrophication.

2. METHODS

Tie Lake was sampled seven times from November 1980 to September 1981 at three stations shown on Figure 1. Each station was located at the deepest point in each of the three basins. Surface and bottom water samples were taken using a Van Dorn bottle and analyzed for nutrients and general water chemistry. Dissolved oxygen and temperature profiles were done using a YSI Model 54 dissolved oxygen meter. In addition a composite sample from each meter of depth was taken for phytoplankton identification and enumeration, Chlorophyll a determinations were done at each meter of depth as an estimate of standing crop.

Water quality samples were also taken on each of five inflow streams and the outflow stream from March 1981 to October 1981 (Figure 1). At the time of sampling, stream flow estimates were done using a Mead model HP-302 velocity meter and the stream cross-section method.

Finally, lake sediment samples were taken once at each station on September 9, 1981 using an Ekman grab.

All chemical and biological analyses were performed by

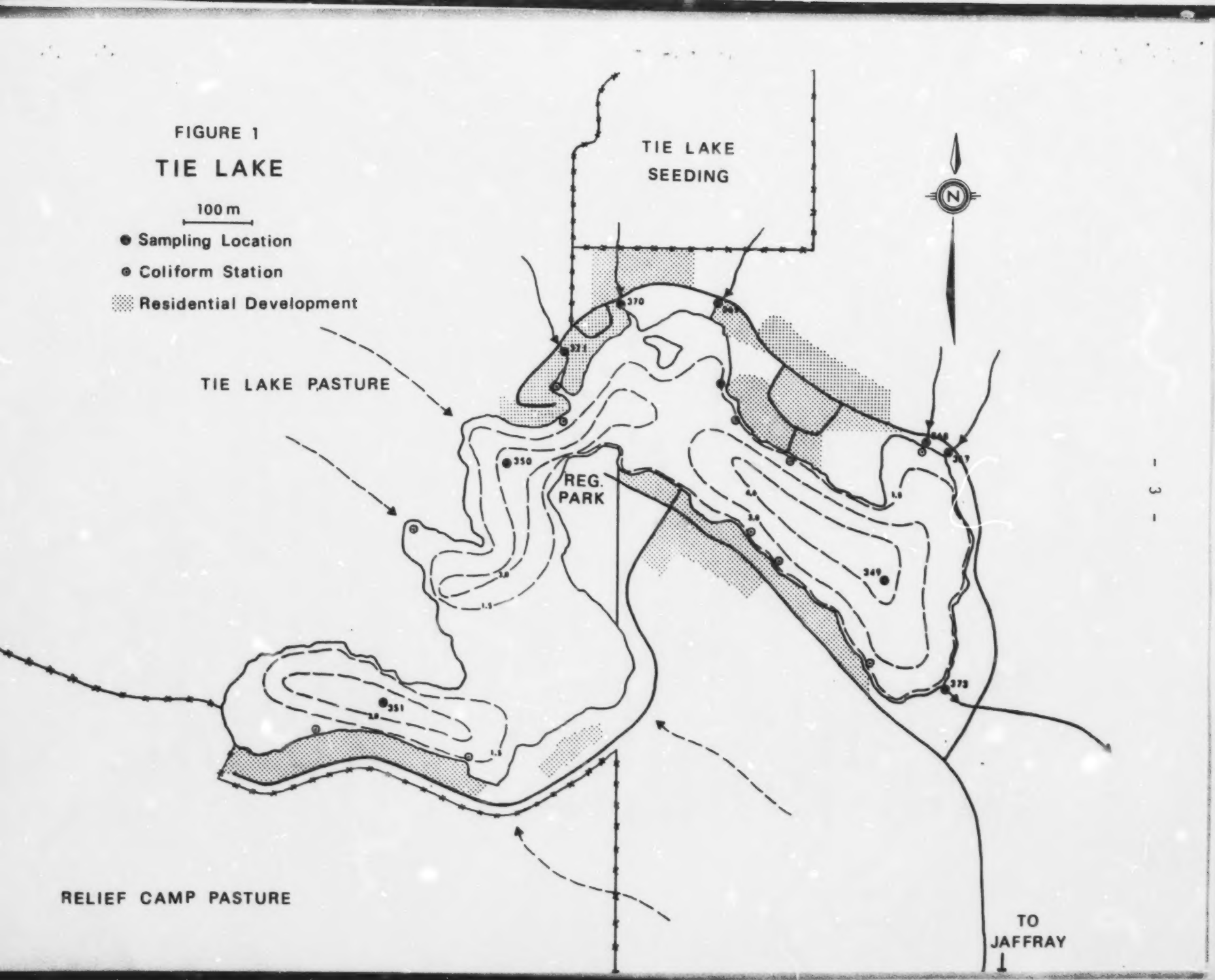
FIGURE 1
TIE LAKE
100 m
Sampling Location
Coliform Station
Residential Development
TIE LAKE SEEDING
TIE LAKE PASTURE
REG. PARK
RELIEF CAMP PASTURE
TO JAFFRAY
N
370
321
350
349
351
373
3.0
1.5
4.0
1.0

the B.C. Ministry of Environment, Environmental Laboratory (B.C. Dept. of Env. 1976). Estimates of runoff to the lake were done by Mr. Larry Barr of the B.C. Water Management Branch, Surface Water Section using field stream guaging, data from nearby watersheds guaged by the Water Survey of Canada, and annual precipitation data from the Cranbrook Airport. Estimates of phosphorus loadings from domestic and livestock sources were done by Dr. John Weins of the B.C. Ministry of Environment, Surveys and Resource Mapping Branch and are contained in a separate report (Weins 1983). Background data for this report were obtained from several sources; pasture use information was supplied by Mr. Ken Gibbard of the B.C. Ministry of Forests, Range Section, while the numbers and distribution of lakeside housing was determined by combining information obtained from the Tie Lake Ratepayers' Association, the Regional District of East Kootenay and the Assessment Authority of British Columbia.

3. RESULTS

3.1 Temperature and Dissolved Oxygen

Being a shallow lake with a mean depth of approximately 2 meters. Tie Lake seldom experiences thermal stratification. During the ice free period the water column is essentially isothermal and well mixed by wind as indicated by similar bottom and surface temperature and concentrations of various chemical parameters. Ice cover begins forming in mid-November and normally disappears by the end of March. Ice regularly reaches a thickness of 30 to

50 cm and thicker during particularly cold winters. Under ice cover a reverse thermal stratification typically develops with less dense water from 0 to 4°C just under the ice overlaying water of maximum density (near 4°).

Figure 2 shows a series of temperature/dissolved oxygen profiles for station 349 in the east basin. The November 20, 1980 profile was taken about two weeks after ice formation while some areas, notably off the north shore point in the east basin, were still as yet unfrozen. These areas are described by local residents as "springs" and are important evidence of groundwater inflow. On this date thermal stratification had not developed although oxygen depletion of bottom water had commenced. On February 25, 1981 water below 2 meters was essentially anoxic and the maximum concentration just under the ice was only 2 mg/L. These levels were sufficiently low to cause a winter kill of fish (Wetzel 1975) although no evidence exists for such an event during the winter of 1981. In the central basin, station 350, there was 4 mg/L dissolved oxygen immediately under the ice dropping to 1.9 mg/L at one meter depth and anoxic conditions from 3 meters to the bottom.

Unfortunately on February 25 the temperature recording instrument was not functioning, however Figure 3 shows a profile for January 19, 1983. This profile illustrates well the formation of winter thermal stratification and deep water oxygen depletion. Oxygen levels decreased with depth from

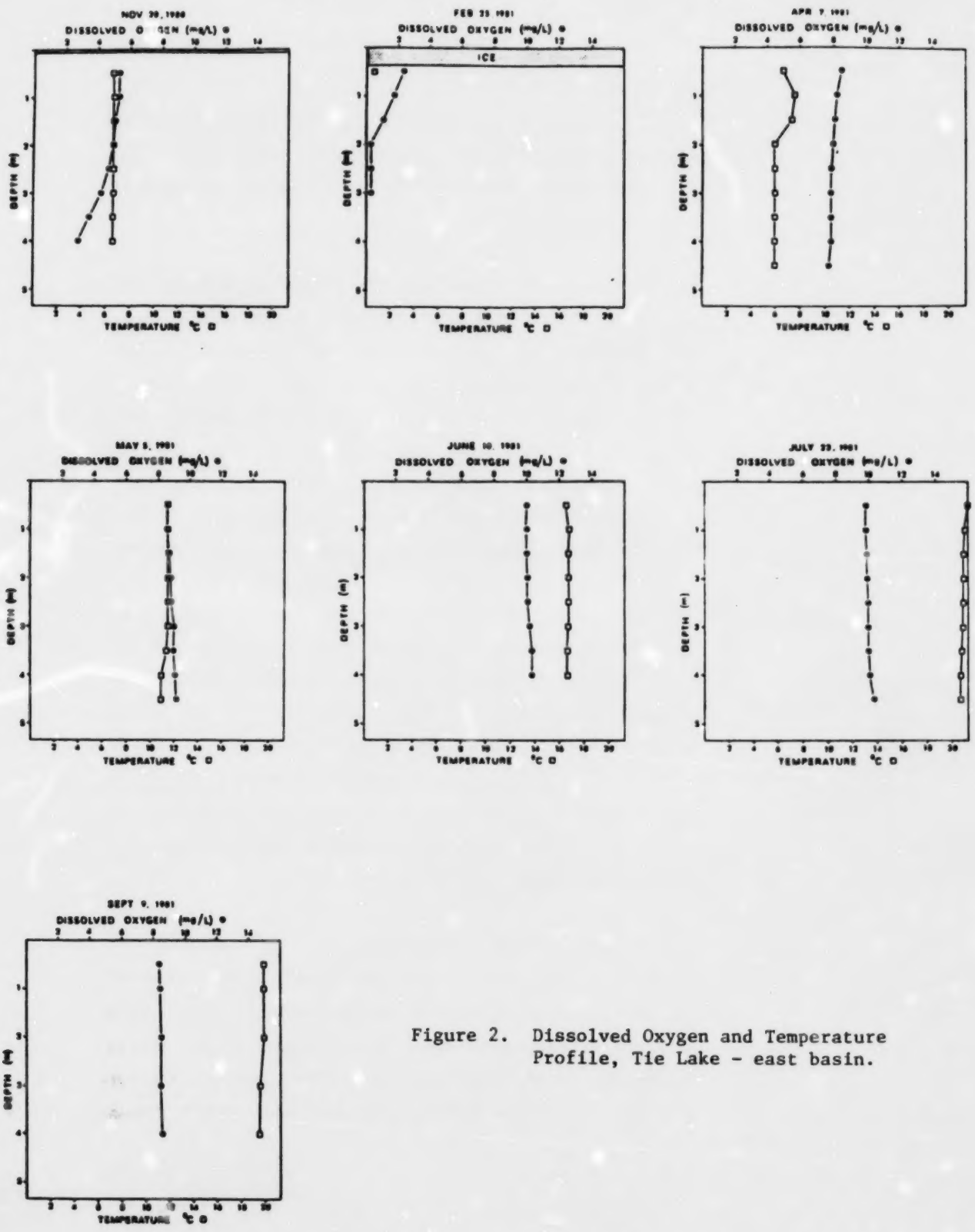

Figure 2. Dissolved Oxygen and Temperature Profile, Tie Lake - east basin.

a maximum just under the ice of 7.3 mg/L (56% saturation) to 1.7 mg/L (14% saturation) at 3 meters. Oxygen levels were not as low as they were in 1980 - 81 which may or may not have been the result of lower productivity and may be a result of instrument error, different climatic factors or the duration of ice cover at the time of sampling. Buchanan and Goddard (1976) show evidence of slight increases in temperature in bottom water and hypothesized the existence of geothermal heating and or groundwater inflow. The 1983 profile did not detect this phenomena although sampling was done to a depth of 3 meters only, the bottom being at about 4 meters. The early winter open water "springs" previously mentioned tend to corroborate the groundwater inflow hypothesis and suggest the formation of horizontal density currents, the significance of which will be discussed later.
(Section 4.1)

FIGURE 3

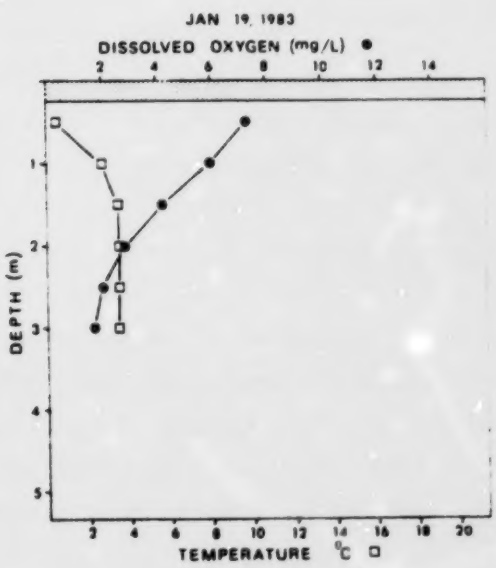

Figure 3. Temperature and dissolved oxygen profile at Station 349 - east basin. Jan. 19, 1983.

Throughout the rest of the study period (Figure 2) little or no thermal gradient was observed as wind action kept the shallow water column well mixed. Buchanan and Goddard (1976) did observe weak stratification during the summer of 1973 following a period of hot, calm weather. From April to July the lake continued to warm to a maximum of about 22°. Tie Lake regularly reaches these high temperatures in the summer with maximums as high as 25° having been recorded (Buchanan and Goddard 1976). Overall water temperatures in the central and west basins were often 1 - 2° warmer than the east basin. The reason for this difference is not known but is not thought to be significant.

Dissolved oxygen levels also failed to show any significant gradients and remained fairly constant from April to September. A slight increase in lake dissolved oxygen from 8 mg/L in May to 10 mg/L in June may or may not have been related to the algal die-off and clearing of the water. Slight differences in dissolved oxygen levels between basins was often observed, though as with temperature was not considered to be significant.

3.2 General Water Chemistry

Table 1 lists results of a number of water quality parameters sampled throughout the study period. These values have been averaged for all stations and all depths, there being no significant spatial trends except for weak stratification under ice on

February 25. Unfortunately the analysis of several parameters was discontinued early in the study. Normally of minor importance with regard to productivity and the trophic status of a lake it will be seen later in the report that dissolved calcium and other indicators of the calcium carbonate equilibrium, alkalinity and hardness, were very important in Tie Lake.

3.2.1 pH

The pH in Tie Lake during the study ranged from 8.0 - 8.6 which is typical of similar hardwater lakes of the region such as Wasa Lake (Crozier 1978) and Edwards Lake (Crozier 1980). There was an obvious trend towards increasing pH throughout the study which, as will become apparent, was one indicator of chemical changes taking place within the lake.

3.2.2 Specific Conductance

This parameter is an indirect measure of the quantity of dissolved salts and in bicarbonate-type lakes (hardwater) is roughly proportional to the concentrations of the major ions (Wetzel 1975). As such the specific conductance in Tie Lake over the study period shows some interesting trends. As with the concentrations of all ions, the greatest value was obtained on February 25 under 50 cm of ice cover. These increases in ionic concentrations are no doubt simply a function of reduced lake volume

Table 1. Concentrations of Various Chemical Parameters: Lake Averages

ND - not done

Date	pH	Specific Conductivity µs/cm	Total Alkalinity mg/L	Hardness mg/L	Total Organic Carbon mg/L	Total Inorganic Carbon mg/L	Silica mg/L	Calcium mg/L	Magnesium mg/L
Nov. 20, 1980	8.	417.	225.	209.	ND	ND	23.4	38.3	27.5
Feb. 25, 1981	8.	453.	244.	228.	16.	65.	24.3	43.7	28.9
Apr. 7, 1981	8.1	405.	ND	ND	ND	ND	ND	ND	ND
May 5, 1981	8.3	410.	ND	ND	10.	55.	16.5	ND	ND
June 10, 1981	8.3	379.	ND	ND	10.	49.	ND	ND	ND
July 23, 1981	8.5	336.	ND	ND	8.	44.	ND	ND	ND
Sept. 9, 1981	8.6	353.	ND	ND	5.	50.	ND	ND	ND
Mean	8.2	393.	234.	218.	10.	53.	21.4	41.	28.2

through the formation of ice. Freezing water tends to exclude dissolved materials (Barisa 1977,Canfield et al 1983) and 50 cm of ice over the entire surface of Tie Lake accounts for a significant 25% of the total lake volume.

Spring values for specific conductance were about the same as the previous fall, however, a sharp drop was recorded on June 10 and a further reduction occurred during July. This indicates either the removal of a significant quantity of dissolved salts from the water column or an increase in the lake volume as the result of the influx of a large quantity of water low in dissolved salts. The rationale for selecting the former is be discussed in Section 4.2.

Fall values increased somewhat over the summer minimum indicating a return to higher levels recorded the previous fall. This suggests the existence of a cyclic process.

3.2.3 Total Alkalinity and Hardness

Both parameters are indicators of the carbon dioxide-bicarbonate-carbonate equilibrium system. Increased quantities of bicarbonate and carbonate cause an increase in total alkalinity which is expressed as $CaCO_3$ equivalents. Inversely, decreases in these ions primarily as a result of precipitation will result in a drop in total alkalinity.

Levels of hardness will change in a similar fashion but only indirectly as hardness is primarily caused by calcium and magnesium, largely combined with bicarbonates and carbonates.

Alkalinity values, only measured during winter, were similar to those found by Buchanan and Goddard in 1972 and 1973. The values for both parameters are similar to other temperate hardwater lakes (Murphy et al 1983, Crozier 1978).

Had these parameters been monitored throughout the study they would have most certainly shown a marked decrease in June similar to that found by Buchanan and Goddard in August 1973. The precipitation of $CaCO_3$, thought to have occurred between May 5 and June 10, would have caused this sharp drop in alkalinity and hardness. Further discussion on the topic of calcite precipitation and its importance in Tie Lake follows.

3.2.4 Total Organic and Inorganic Carbon

Along with phosphorus and nitrogen, carbon is one of the most essential nutrients for algal growth. The carbon:nitrogen:phosphorus ratio in plant tissues is about 40C:7N:1P (Wetzel 1975) but because carbon is so abundant in many forms it is rarely in short supply although its requirements by weight are highest. The C:N:P ratio in Tie Lake averaged about 3250:93:1 over the five samples for which all three parameters were monitored. (Table 1) This gives some idea of the abundance of carbon, and for that matter nitrogen, in relation to phosphorus.

Total inorganic carbon (TIC) is essentially the sum of the components of the carbon dioxide-bicarbonate-carbonate equilibrium system. Inputs to this system include the weathering of minerals such as limestone transported to the lake via inflowing water and atmospheric CO_2 solubility. Exports include biotic uptake or conversion to organic forms and $CaCO_3$ precipitation and lake water outflow.The June and July samples were significantly lower than the spring levels, likely a result of calcite precipitation. Accumulation of inputs caused the September sample to increase to near spring levels.

Total organic carbon (TOC) exists in a very complex array of forms but is usually broken into dissolved (DOC) and particulate (POC) forms. POC is comprised of detritus and a very small portion in living zooplankton and phytoplankton. DOC includes allochtonous forms (originating from outside the lake) comprised of humic and fulvic acids and autochtonous forms (processed within the lake) which consist of amino acids, fatty acids and other cellular by-products (Wetzel 1975). POC typically makes up only about 10% of the TOC with a much smaller portion as living POC so that changes in the plankton communities do not greatly effect TOC levels (Wetzel 1975). This is supported in Tie Lake by residue analyses which give an indication of organic carbon. Non-filterable volatile solids (approximately equivalent to POC) were consistently about 3 mg/L (n=4) while filterable volatile solids (approximately equivalent to DOC) were about 120 mg/L before May 5 and about 100 mg/L after (n=4).

It is thought that Tie Lake receives fairly high levels of allochtonous DOC in the form of humic acids from inflow streams and overland flow. DOC (from residue analyses) in inflows ranged from 43 -135 mg/L (n=4) and many samples had a brown tint indicative of humic acids. These substances may impart a brown colour to the lake water which would be removed by calcite precipitation as humic acids and other organic compounds are strongly adsorped to $CaCO_3$ (Wetzel 1975, Jorgensen 1980). This may explain the drop in TOC through the study period although the processes involved are complex.

Both TIC and TOC levels are similar to those found by Buchanan and Goddard. TIC levels in 1973 showed a similar decrease after August calcite precipitation however TOC levels increased indicating other factors are involved.

3.2.5 Dissolved Silica

This nutrient is essential for the growth of diatoms, usually the dominant phytoplankton group in oligotrophic lakes. Levels of dissolved silica may reach very low levels during spring diatom blooms and can limit growth (Wetzel 1975). Dissolved silica levels below 0.5 mg/L are usually considered limiting to diatom growth (Trainor 1978). The spring dissolved silica concentration in Tie Lake was 16.5 mg/L (Table 1). As will be discussed later, other factors control diatom growth and prevent this group from dominating the phytoplankton. (Section 3.4)

3.2.6 Dissolved Calcium and Magnesium

Together these ions are the main cause of hardness in water and levels are typical of the fairly hardwater lakes of the region. Calcium, particularly in Tie Lake, is singularly important during calcium carbonate (calcite) precipitation because the solid which forms strongly adsorbs, among other things, phosphorus and transports it to the lake sediments.

Levels of both calcium and magnesium are similar to that found by Buchanan and Goddard though it is unfortunate that calcium was not monitored throughout the study. Most certainly the June 10 sample would have shown a sharp decrease similar to that found in August 1973 when the concentration dropped from 40 mg/L to 19 mg/L as a result of calcite precipitation. Magnesium levels in August 1973 were similar to those found in 1980 - 81 because calcite formation does not affect this ion.

3.2.7 Faecal Coliform Organisms

This test is a quantitative enumeration of a group of bacteria associated with warm-blooded bird and mammal faeces. Samples were taken at various times and in various locations along the shore to determine the presence of sewage from lakeside septic systems. (See Figure 1) Individual sources of contamination could not be determined from the sampling and in fact, positive results could be the result of livestock

grazing nearby or even waterfowl. Out of a total of 15 samples, 11 were positive with a maximum of 22 MPN/100 ml. Most counts were in the 2 -5 MPN/100 ml range. Two of these samples were what might be called controls being located in remote areas away from human influence and one of these was positive. May and June sampling, which account for 9 positive results, were possibly influenced by huge numbers of waterfowl seen on the lake. There is some evidence, however, although far from conclusive, to suggest contamination by lakeside septic systems. This is important from a nutrient standpoint but it should be noted that levels are within recommended health standards for primary contact recreational use (swimming, water skiing, etc.).

3.3 Nutrients

3.3.1 Nitrogen

Nitrogen is generally considered second to phosphorus in importance in determining levels of productivity in lakes. This is simply due to the fact that in this region it is seldom in as short supply as phosphorus, although in rare instances has been found to be limiting. Nitrogen exists in several different forms, each a component of an oxidation series resulting from aerobic heterotrophic bacterial decomposition of organic matter. The end product of this series is nitrate (NO_3) and is generally the most plentiful form in oligotrophic systems where

organic matter is low and oxygen levels are high. Table 2 shows that in Tie Lake most of the nitrogen exists in organic form or as ammonia. Over the course of the study these two forms accounted for about 95% of the total nitrogen in the lake.

The organic nitrogen fraction is comprised of living phyto and zooplankton and decomposing organic matter from plankton, littoral vegetation and terrestrial sources. Levels of this form of nitrogen are always high in Tie Lake (Table 2, Buchanan and Goddard 1976) but were at their highest during the bloom of 1980-81.

The other major form of nitrogen in Tie Lake was ammonia which is the end product of heterotrophic bacterial decomposition of organic matter. Ammonia is also the major excretory product of zooplankton though this source is minor by comparison (Wetzel 1975). Ammonia showed a much more dramatic temporal trend through the study than organic nitrogen reaching levels up to 2 mg/L under the ice and dropping to about 5% of this level by September (Table 2 and 3). Levels as high as 4 to 5 mg/L were recorded by Buchanan and Goddard in 1972 and 1973.

Ammonia in water exists as an equilibrium between ammonium ion (NH_4^+) and unionized ammonia (NH_3). The dissociated NH_4^+ is by far the predominant form but the unionized NH_3 is highly toxic to fish. The relative proportions of each are dependent on pH and temperature, with the unionized fraction increasing

Table 2. Concentrations of Various Forms of Nitrogen in Tie Lake: By Station and Depth. E - East Basin, C - Central Basin, W - West Basin; S - Surface, B - Bottom

Date		Organic Nitrogen mg/L			Total Ammonia mg/L			Nitrite µg/L			Nitrate mg/L			Total Nitrogen mg/L		
		E	C	W	E	C	W	E	C	W	E	C	W	E	C	W
Nov. 20, 1980	S	2.			1.16			11.			0.06			3.23		
	B	2.			1.34			15.			0.05			3.4		
Feb. 25, 1981	S	1.	2.		1.57	1.6		L5.	L5.		0.02	0.03		2.59	3.63	
	B	2.	2.		1.85	1.91		L5.	L5.		0.07	L.02		3.92	3.91	
Apr. 7, 1981	S	1.	2.	2.	1.57	1.24	1.38	6.	10.	5.	L.02	0.06	L.02	2.58	3.98	3.38
	B	1.	2.	2.	1.51	1.45	1.4	6.	8.	5.	L.02	0.03	L.02	2.52	3.56	3.4
May 5, 1981	S	2.	1.	1.	1.01	0.96	.895	34.	29.	21.	0.16	0.16	0.14	3.2	2.15	2.06
	B	1.	1.	1.	1.02	0.95	.96	37.	27.	21.	0.15	0.17	0.2	2.21	2.15	2.18
June 10, 1981	S	2.	2.	1.	0.475	.386	.564	7.	6.	7.	0.13	0.1	0.1	2.61	2.49	1.67
	B	2.	1.	1.	0.484	.374	.576	7.	6.	7.	0.12	0.1	0.1	2.61	1.48	1.68
July 23, 1981	S	0.92	1.07	0.88	0.102	.093	.077	L5.	L5.	L5.	0.04	0.03	0.03	1.06	1.19	0.99
	B	0.94	0.97	1.01	0.095	.087	.075	L5.	L5.	L5.	0.03	0.02	0.03	1.06	1.08	1.11
Sept.9, 1981	S	1.01	0.95	0.92	0.08	.054	.028	L5.	L5.	L5.	L.02	L.02	L.02	1.09	1.00	0.95
	B	0.97	0.92	1.05	0.079	.068	.033	L5.	L5.	L5.	L.02	L.02	L.02	1.05	0.99	1.08
Mean	S	1.42	1.50	0.97	0.85	0.72	0.49	8.3	7.5	5.5	0.06	0.06	0.04	2.34	2.41	1.51
	B	1.42	1.31	1.01	0.91	0.81	0.51	9.3	6.8	5.5	0.06	0.05	0.05	2.68	2.19	1.57

Table 3. Concentrations of Various Forms of Nitrogen: Lake Averages and Percentage of Total Nitrogen

Date	Org. N		Total Ammonia		Nitrite		Nitrate		Total Nitrogen	
	mg/L	%	mg/L	%	µg/L	%	mg/L	%	mg/L	%
Nov. 20, 1980	2.	60.	1.25	38.	12.	0.4	0.05	1.5	3.31	100
Feb. 25, 1981	1.7	48.	1.73	49.	L5.	0	0.03	1.2	3.51	100
Apr. 7, 1981	1.7	52.	1.42	44.	6.7	0.2	0.02	0.6	3.24	100
May 5, 1981	1.2	52.	0.96	41.	28.	1.2	0.16	6.9	2.32	100
June 10, 1981	1.5	72.	0.48	23.	6.7	0.3	0.11	5.3	2.09	100
July 23, 1981	0.96	89.	0.09	8.3	L5	0	0.03	2.8	1.08	100
Sept. 9, 1981	0.97	94.	0.06	5.8	L5	0	L.02	0	1.03	100
Mean	1.4	59.	0.85	36.	7.6	0.3	0.06	2.5	2.37	100

in warmer, alkaline conditions. Calculations of the unionized NH_3 fraction of the total ammonia were determined using a relationship presented by Trussell (1972) and are listed in Table 4.

TABLE 4 Unionized ammonia levels in Tie Lake 1980-81 (µgN/L) S-surface, B-bottom

	STATION						
	EAST		CENTRAL		WEST		LAKE AVERAGE
DATE	S	B	S	B	S	B	
Nov.20/80	18	16	ND		ND		17
Feb.25/81	24	42	25	19	ND		27.5
Apr.7/81	24	15	19	22	26	27	22.2
May 5/81	25	25	24	29	37	30	28.3
June 10/81	21	17	23	22	34	29	24.3
July 23/81	10	9	7	7	7	7	7.8
Sept.9/81	7	5	4	5	2	3	4.3

Levels were highest during the winter and spring reaching a maximum value of 42 µg/L on February 25 in the east basin. Pommen (1983) summarizes the toxic effects of unionized ammonia on fish and found that lethal concentrations lie between 0.2 and 0.7 mg N/L based on 1 to 4 day LC_{50} values. Far lower values of 25 µg N/L have been shown to be harmful to fish under continuous exposure. The latest water quality criteria for un-ionized ammonia is a maximum of 30 µgN/L and a continuous exposure (more than 10 days) maximum of 7 µgN/L (Pommen, pers. comm.). Recent research, which has led to a critical review of the EPA un-ionized ammonia toxicity criteria of 50 µg/N/L

indicates that toxicity may be reduced at a higher pH despite the fact that the concentration will rise due to a shift in the equilibrium This apparently is because the toxicity of any given concentration of un-ionized ammonia may be greater or less depending on the local water chemistry. Put another way, the toxicity and the concentration are not necessarily proportional.

In any event there is no evidence the elevated levels of un-ionized ammonia found in Tie Lake in 1981 resulted in any fish kills. To understand the true impact of this parameter on the lake's population, either lethal or sub-lethal, will require more study.

Nitrite (NO_2) is a transitional ion between ammonia and nitrate and is rarely found in significant quantities usually being rapidly oxidized to nitrate. Sporadic detectable levels of nitrite were found in Tie Lake particularly on May 5 when a maximum of 34 µg/L occurred (Table 2). This ion is also highly toxic to fish and 96 hr. LC_{50} values as low as 0.11 mgN/L NO_2 have been found (Pommen, unpubl). Levels in excess of 20 µgN/L are found to create stress in fish with continuous exposure. It can be seen from Table 2 that during May 1981 levels of nitrite exceeded this level but remained below the recommended maximum acutely toxic level of 60 µgN/L (Pommen, 1983).

Nitrate levels in Tie Lake were uniformly low throughout the study with a maximum concentration of

0.16 mg/L on May 5 (Table 2) and never forming more than 7% of the total nitrogen (Table 3). Buchanan and Goddard found higher levels of nitrate in May 1972, throughout the rest of their study, however, levels remained very low or undetectable. Nitrate is not normally toxic to fish.

Figure 4 illustrates the changing concentrations of each form of nitrogen in the lake throughout the study. It is not clear why ammonia, in the presence of dissolved oxygen, does not nitrify to nitrite. Elevated ammonia levels in bottom samples in November and February (Table 2) under anaerobic conditions indicate that there may have been some release from bottom sediments. The steady decline of ammonia concentrations through the spring and summer may be due to sedimentation. In alkaline lakes with high levels of humic acids, ammonia is strongly sorbed to particles and colloids (Wetzel 1975). This material may in turn be sorbed to calcite particles during calcium carbonate precipitation and readily settled out. The drop in ammonia levels does not however seem strongly correlated to the calcite precipitation in May, with falling concentrations being detected before and after. To further speculate it is possible that over the spring and summer a portion of the ammonia volatilized and was lost to the atmosphere. This volatilization would be encouraged by the steadily increasing pH (Table 1) which would shift the equilibrium from ionized to un-ionized ammonia.

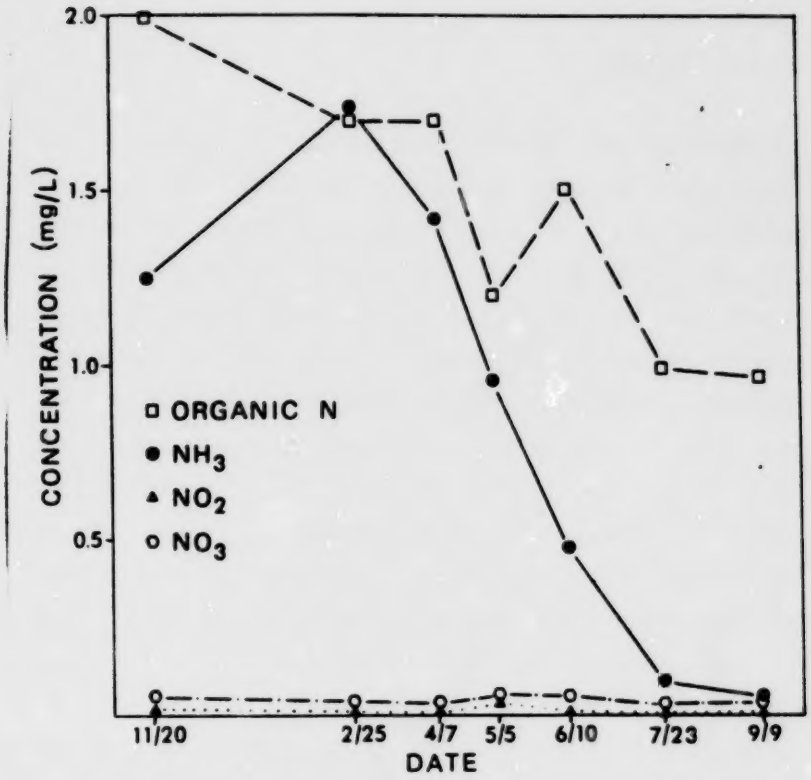

Figure 4. Fractions of nitrogen in Tie Lake 1980 - 81 (Lake Averages)

The possibility that nitrification is complete and that nitrate is rapidly utilized by algae does not seem likely. First, the uptake of nitrate by algae is inhibited in the presence of ammonia (Jorgensen 1978). Second, the highest levels of nitrate would be expected during periods of low algal production yet Table 3 shows the highest levels in May when phytoplankton production was high. In addition, nitrogen in the small inflow streams, unaffected by phytoplankton activity was comprised almost entirely of organic forms (90 - 100%) with the remainder as ammonia. Nitrite and nitrate were never detected in these inflow streams.

3.3.2 Phosphorus

Phosphorus is genrally considered to be the most importan nutrient in lakes systems due to its scarcity and has most often been found to be the factor which klimits the growth of phytoplankton (Vollenweider and Dillon 1974). Phosphorus in water exists in a complex array of forms that to date are poorly understood. Common phosphorus fractions used in this study (Tables 5 and 6) are based on analytical procedure rather than on chemical classifications. The significance of each of these fractions is a matter of continued debate among contemporary researchers. The forms of phosphorus being referred to are total phosphorus, total suspended phosphorus (TSP) also referred to as particulate phosphorus including both organic and inorganic forms and total dissolved phosphorus (TDP) which also includes organic and inorganic forms. The TDP is further broken into soluble reactive phosphorus (SRP) or dissolved orthophosphate, the simplest most rapidly utilized form, and dissolved organic phosphorus (DOP) determined by subtracting SRP from TDP. In relating phosphorus levels to trophic status and for predictive purposes this report uses total phosphorus primarily because empirical relationships have proven it reliable and no other fraction has been demonstrated as having a higher correlation to lake phytoplankton productivity (Peters and Bergmann 1982).

Table 5 shows that SRP levels were rarely over the detectable limit of 3 µgP/L (n=36), a situation also recorded by Buchanan and Goddard in 1973. In eutrophic lakes SRP will rarely accumulate because it is utilized so rapidly. Most of the dissolved

phosphorus then exists as organic compounds so that the DOP and TDP in Tables 5 and 6 are virtually the same. The TDP concentrations reached a maximum under ice and remained elevated until the May sample when dropped by about 40%. During the bloom period in 1980 - 81 TDP accounted for 27 -48% of the total phosphorus, after calcite precipitation this increased to about 60%. (Table 6)

TSP or particulate phosphorus levels were more seriously affected by calcium carbonate precipitation, falling by about 60% from the May to June samples (Table 6). TSP represented 50 - 70% of the total phosphorus prior to calcite formation and 45-50% after. It may be that phosphorus in the seston, already in particulate form, is more readily settled out when sorbed to $CaCO_3$ than are the dissolved forms simply by virtue of the larger resulting agglomerate.

Total phosphorus levels in Tie Lake during spring overturn in 1981 were about 30 µg/L. This level remained fairly constant from the previous fall with a maximum of 40 µg P/L occurring in February. After calcite precipitation in May the total phosphorus concentration decreased by about 50% to 14 µg P/L and remained at this level for the rest of the study. This dramatic decrease in phosphorus after $CaCO_3$ precipitation was not recorded by Buchanan and Goddard in 1973 although levels were never as high as 1980-81. Spring overturn levels in 1972 and 1973 were one third to one half that of 1981 which suggests an increase in phosphorus loading to the lake over the eight year period.

Table 5. Concentrations of Various Forms of Phosphorus in Tie Lake by Station and Depth.

E - East Basin, C - Central Basin, W - West Basin; S - Surface, B- Bottom.

Date		Soluble Reactive Phosphorus (µg/L)			Dissolved Organic Phosphorus (µg/L)			Total Dissolved Phosphorus (µg/L)			Total Suspended Phosphorus (µg/L)			Total Phosphorus (µg/L)		
		E	C	W	E	C	W	E	C	W	E	C	W	E	C	W
Nov. 20, 1980	S	3			5			8			22			30		
	B	L3			8			8			21			29		
Feb. 25, 1981	S	4	L3		16	12		20	12		10	45		30	57	
	B	L3	L3		19	15		19	15		15	27		34	42	
Apr. 7, 1981	S	L3	L3	L3	15	14	13	15	14	13	18	11	14	33	25	27
	B	L3	L3	3	13	15	13	13	15	16	19	13	19	32	28	35
May 5, 1981	S	L3	L3	7	12	13	11	12	13	16	16	13	13	28	26	29
	B	L3	L3	L3	12	16	9	12	16	9	17	28	17	29	44	28
June 10, 1981	S	L3	L3	L3	8	9	9	8	9	9	4	5	7	12	14	11
	B	L3	L3	L3	8	9	11	8	9	11	10	4	8	18	13	18
July 23, 1981	S	L3	L3	L3	7	7	8	7	7	8	6	7	9	13	14	15
	B	L3	L3	L3	7	7	9	7	7	9	5	7	9	12	14	14
Sept. 9, 1981	S	L3	L3	L3	8	8	7	8	8	7	4	5	8	12	13	12
	B	L3	L3	L3	8	8	9	8	8	9	6	5	8	14	13	14
Mean	S	L3	L3	1.4	10.1	10.5	9.6	11.1	10.5	10.6	11.4	14.3	10.2	22.6	24.8	18.8
	B	L3	L3	L3	10.7	11.7	10.2	10.7	11.7	10.8	13.3	14	12.2	24	25.7	21.8

Table 6. Concentrations of Various Forms of Phosphorus in Tie Lake: Lake Averages and Percentage of Total Phosphorus

Date	S.R.P. µg/L	S.R.P. %	D.O.P. µg/L	D.O.P. %	T.D.P. µg/L	T.D.P. %	T.S.P. µg/L	T.S.P. %	T.P. µg/L	T.P. %
Nov. 20, 1980	L3	L1	6.5	22	8	27	21.5	73	29.5	100
Feb. 25, 1981	L3	L1	15.5	38	16.5	40	24.2	59	40.7	100
Apr. 7, 1981	L3	L1	13.8	46	14.3	48	15.7	52	30	100
May 5, 1981	L3	L1	12.2	40	13	42	17.3	56	30.7	100
June 10, 1981	L3	L1	9	63	9	63	6.3	44	14.3	100
July 23, 1981	L3	L1	7.5	55	7.5	55	7.2	52	13.7	100
Sept. 9, 1981	L3	L1	8	61	8	61	6	46	13	100
Mean	L3	L1	10.3	42	10.9	44	14	57	24.5	100

According to Wetzel (1975) a total phosphorus concentration of 30 µg P/L classifies Tie Lake on the borderline between meso-eutrophic (10 - 30 µg P/L) and eutrophic (30 - 100 µg P/L). This means that phytoplankton growth may periodically reach levels high enough to produce undesirable lake water quality. Problems associated with high levels of phytoplankton growth include a green, opaque appearance with low transparency, a fishy or grassy odour and accumulated oxygen depletion as the algal biomass decomposes. The relationship between phosphorus levels and phytoplankton growth or productivity during this study is illustrated in Figure 5. Chlorophyll a, an algal pigment, is an estimate of the algal standing crop and the dependence on total phosphorus concentration is clearly evident. Further evidence that phosphorus levels control and limit phytoplankton growth in Tie Lake is found in the ratios of total nitrogen to total phosphorus in the water column. Based on the respective metabolic requirements of each nutrient, an N:P ratio greater than 15:1 is considered to be indicative of phosphorus limitation (Nordin pers.comm.). N:P ratios in Tie Lake throughout the study ranged from a low of 49:1 to a high of 217:1 with a mean of 99:1 (n=36) which indicates a strongly phosphorus limiting system.

As a final observation on phosphorus, an examination of Table 5 does not indicate any distinct spatial trends between basins or depths. Elevated bottom levels, particularly dissolved forms, might be expected from the February sample as a result of

phosphorus release from anaerobic sediments but this did not appear to be the case. The significance of this apparent lack of sediment phosphorus release will be discussed in more detail during the calculation of the phosphorus loading.

3.4 Phytoplankton Composition and Standing Crop

Figure 6 shows the change in Secchi depth over the study period. This parameter is a simple measure of the transparency of water which can be severely reduced during heavy phytoplankton blooms and to a lesser extent in the presence of coloured compounds in the water. The dramatic increase in Secchi depth on June 10, essentially to the bottom at 4 - 5 meters, is thought to be associated with calcite precipitation. A reduction in the Secchi depth in September indicates a slight increase in productivity. The increase in chlorophyll levels (Figure 5) at this time however were very minor and the reason for increased turbidity is not clear.

Chlorophyll a concentrations, a direct quantification of phytoplankton biomass followed a similar trend (Figure 5), with a marked decrease in June. The increased concentration in February, over November and April, was likely due to reduced lake volume as a result of ice formation, as rather than a true increase in algal biomass.

Phytoplankton identification and enumeration was also done to determine changes in population size and composition and the dominant species. Table 7 shows the relative proportions for each major taxonomic group based on cell counts. Under each phylum or sub-phylum the most abundant genera are listed

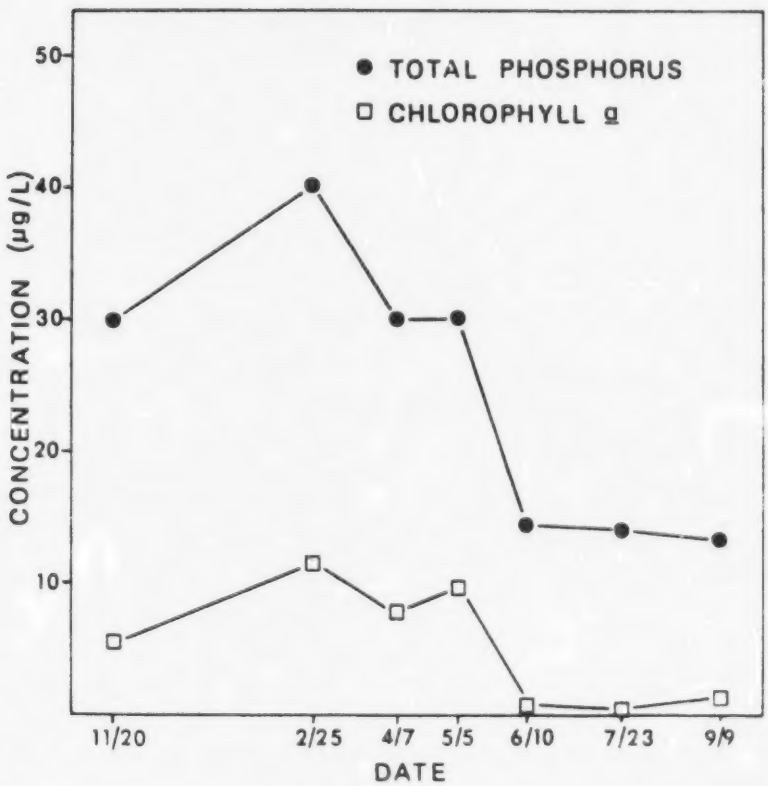

Figure 5. Total Phosphorus and Chlorophyll a at Tie Lake 1980 - 81 (Lake Averages)

roughly in a descending order. The 1980 bloom was dominated by the colonial blue-green Microcystis aeruginosa which comprised 95% of the total population. Some confusion arose during the study in distinguishing between Microcystis and the similar genus Aphanocapsa due to preservation artifacts. Some authors have included both genera under the common genus Anacystis (Palmer 1977). This genus continued to dominate through the winter and spring until calcite precipitation in mid May. In June Microcystis still comprised a significant 16% of the phytoplankton although the cryptomonad flagellate Dinobryon and the greens Gloeocystis and Oocystis were slightly more abundant, each representing about 20% of the population. Microcystis again became established in July rising to 77% of the total cell count but fell to 22% in September as the chrysophyte flagellate Dinobryon comprised 32% of the phytoplankton.

The reason that Microcystis and a few similar blue-greens became so dominant in 1980 is unclear but may have been the result of high levels of nitrogen and phosphorus. Blue-greens in general have higher cellular requirements for these nutrients than other algae, in fact Prescott (1968) suggests that 30 µg P/L total phosphorus (coincidentally the spring and fall concentration in Tie Lake in 1980-81) is a minimum required concentration. Vollenweider (1968) has however suggested that blue-green blooms will occur if the spring phosphorus concentration exceeds 10 µg P/L and the nitrogen concentration exceeds 200 - 300 µg N/L. It has certainly been well established

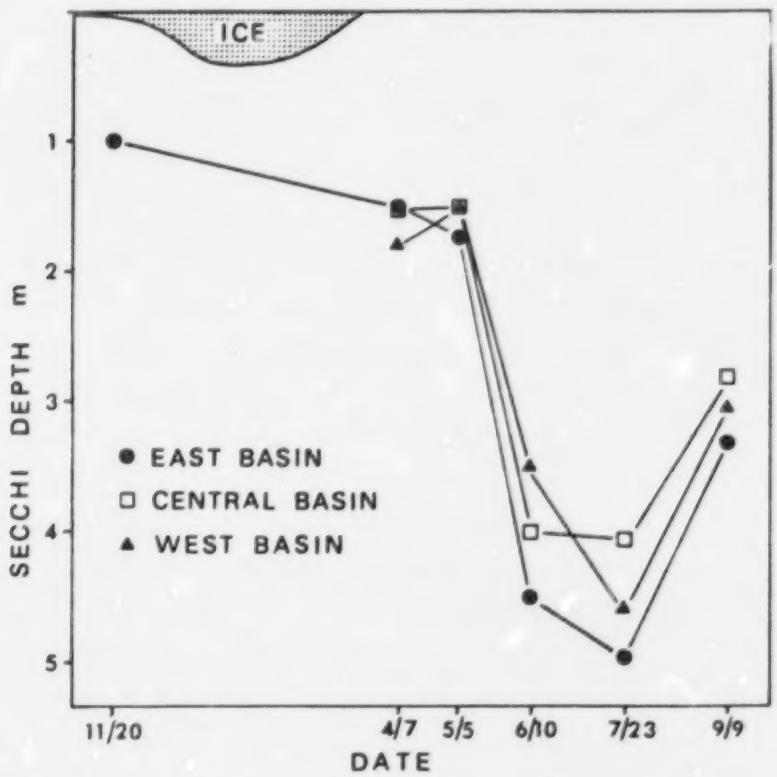

Figure 6. Changes in Secchi depth from November, 1980 to September, 1981 at Tie Lake. Values over 4 m usually indicate that the bottom was visible.

that blue-greens usually abound in shallow, eutrophic, hardwater lakes high in dissolved organic matter such as Tie Lake (Fogg 1973; Trainor 1978; Prescott 1968; Wetzel 1975). In such lakes, particularly with high summer temperatures it is not surprising that diatoms do not fair well despite the abundance of dissolved silica. These algae prefer cold (<15°) deep, oligotrophic lakes where they usually dominate the phytoplankton (Wetzel 1975).

In 1972-73 Buchanan and Goddard found the dominant genera to be the chlorophytes Eudorina and Scenedesmus respectively although Anacystis was reported in 1972. It may have been that in 1980 nutrient concentrations reached high enough levels to allow blue-greens to become established and indeed to grow profusely.

Prescott (1968) suggests that some species of Microcystis, once established as a dominant, may actually produce a factor inhibitory to the growth of other algae which may explain persistence of this alga in Tie Lake through the winter and spring of 1981. After calcite precipitation and associated reduction in nutrient levels other greens and flagellates became more abundant being better adapted to the lower nutrient concentrations.

Microcystis aeruginosa has been known to produce an endotoxin released from dead or dying cells that has lead to poisonings of fish and the death of cattle and waterfowl (Trainor 1978; Prescott 1978). The occurrence of such poisonings is sporadic and the

Table 7. Proportions of Dominant Phytoplankton Taxa in Tie Lake 1980-81 (Percentage of Total Number of Cells/ml)

	80/11/20	81/4/7	81/5/5	81/6/10	81/7/23	81/9/9
Blue-greens (Cyanophyta)	97	51	87	22	88	38
Microcystis (=Anacystis)	95	36	82	16	77	22
Gomphosphaeria		11				
Chroococcus						
Merismopedia						
Lyngbya						
Greens (Chlorophyta)	1	16	7	48	10	25
Gloeocystis				20		14
Oocystis				19		
Elakatothrix						
Botryococcus						
Pediastrum						
Scenedesmus						
Crucigena						
Ankistrodesmus						
Carteria						
Yellow-greens (Chrysophyta)	L1	18	L1	0	L1	3
Dinobryon						3
Mallomonas						
Uroglena						
Diatoms (Bacillariophyceae)	L1	10	4	5	L1	1
Fragilaria						
Nitzschia						
Navicula						
Achanthes						
Cryptophyta	L1	L1	2	24	L1	1
Chroomonas				23		
Cryptomonas						

mechanism not well understood. The presence of M. aeruginosa does not mean that such poisonings would ever occur and no evidence exists to suggest they have occurred at Tie Lake.

3.5 Sediments

Sediment grabs were taken at each of the three lake stations on September 9, 1981 to determine the chemical composition. Table 8 gives a breakdown of the major and minor constituents averaged for the three stations. Carbon was by far the most abundant element indicating a large portion of organic matter. The visual appearance of the sediment, at least to a depth of 30 cm (depth of an Ekman grab), was that of a jelly-like, brown ooze. Large quantities of the macro algae Chara, encrusted with calcium carbonate deposits, were also brought up in the grabs. This alga and large quantities of the abundant littoral macrophytes may contribute substantially to the mass of this sediment during periods of die-off. The Chara no doubt often sinks and dies under the weight of heavy calcite deposits (Wetzel 1975). In shallow areas near shore in the central and east basins this organic sediment does not accumulate and the lake bottom is comprised of the unconsolidated gravels characteristic of the area.

The next most abundant element was calcium which has probably accumulated as a result of periodic $CaCO_3$ precipitation. Through ratios of atomic mass, this calcium carbonate carbon accounts for about 40 mg/g

of the 256 mg/g total carbon. This theorized accumulation of calcium in the sediments through calcite precipitation is supported by comparing the relative proportions of calcium to magnesium in both the sediments and the water column. In the lake water the Ca:Mg ratio is about 1.5:1, while the sediment ratio is 25:1. The reason for this is that magnesium is not affected by calcite precipitation and does not accumulate in sediments to such an extent.

TABLE 8 ANALYSIS OF TIE LAKE SEDIMENT
AVERAGES OF THREE STATIONS

Sampled September 9, 1981.

ELEMENT	µg/g DRY SEDIMENT
CARBON:TOTAL	256700.
CALCIUM	132000.
NITROGEN:KJELDAHL	20400.
MAGNESIUM	5160.
ALUMINUM	2390.
IRON	2360.
PHOSPHORUS:TOTAL	764.
STRONTIUM	179.
BARIUM	144.
MANGANESE	126.
TITANIUM	39.
ZINC	32.
ANTIMONY	31.
LEAD	31.
COPPER	9.
ARSENIC	9.
MOLYBDENUM	6.
VANADIUM	6.
CHROMIUM	4.
CADMIUM	1.
NICKEL	<5.
COBALT	<10.

The nitrogen:phosphorus ratio in the sediment was about 27:1 which was somewhat lower than that found in the lake water (from 50-200:1). This might

suggest a greater sedimentation of phosphorus in proportion to nitrogen though this relationship is unclear. Nitrogen and phosphorus content of littoral vegetation probably affects the sediment N:P ratio to a greater extent and in general no conclusions can be made without further investigation.

Significant levels of barium and strontium in Tie Lake sediments (Table 8) are likely due to the weathering of sedimentary deposits high in alkaline-earth metals such as calcium, magnesium, barium and strontium. All likely accumulate in the lake sediments as a result of precipitation with carbonate ion. Sulphate precipitation can occur but levels of SO_4 in Tie Lake were below the 5 mg/L detection limit (n=4). (McNeely 1979).

4. DISCUSSION

4.1 Dissolved Oxygen Depletion

The depletion of dissolved oxygen in lakes during periods of ice cover is primarily influenced by the sediment surface:lake volume ratio and the lake trophic status with the former accounting for 72 - 78% of the variation in the rate of oxygen depletion (Mathias and Barica 1980). As the sediment surface area:volume ratio increases so does the oxygen depletion rate and a similar effect results from increasing productivity or eutrophication.

Mathias and Barica (1980) found that the sediments of eutrophic lakes consumed oxygen about 3 times faster than those of oligotrophic lakes although this response was a long term one and occasional fluctuations in productivity (blooms) had little effect. This is primarily because, although bacterial oxidation of seston and dissolved organic material does take place in the water column, far more oxygen is consumed at the sediment/water interface (Mathias and Barica 1980; Wetzel 1975).

Using a relationship between sediment surface area:lake volume and winter oxygen depletion rate developed by Mathias and Barica (1980) the oxygen depletion rate for Tie Lake can be estimated. The relationship regression equation is:

$$y = 0.226x + 0.01$$

where y is the oxygen depletion rate in $g/m^3/d$ and x is the sediment surface area:volume ratio in m^2/m^3. This relationship is one developed through the study of a number of eutrophic prairie lakes, many with surface areas and chlorophyll *a* concentrations similar to Tie Lake. A considerable and unknown degree of error no doubt exists in this approach and it is used here merely to illustrate the change in oxygen depletion rate, seen as related to the probability of winter fish kills, produced by altering the mean depth (volume). At a mean depth of 1.9 m, calculated with the excavated outflow channel open and assumed to be a minimum lake level, the oxygen depletion rate would be 0.13 $g/m^3/d$. At a mean depth of 2.4 m, calculated with the culvert as the only outflow plus 20 cm and assumed to be a

maximum lake level to prevent flooding, the oxygen depletion rate would be 0.10 $g/m^3/d$. It can be seen that even with the maximum feasible manipulation of lake level or volume by raising and lowering the outflow the oxygen depletion rate does not change greatly.

Horizontal density currents caused by groundwater inflow, sediment heat transfer or solar radiation may affect the supply of oxygen to the sediments. Initially this would prevent the sediment from becoming anaerobic and possibly releasing adsorbed nutrients under reducing conditions. However, Mathias and Barica (1980) found that below an average lake oxygen concentration under ice of 3.8 mg/L (not calculated for Tie Lake) the oxygen depletion rate begins to drop due to oxygen limitation. Presumably increasing the supply of oxygen to the sediment interface through density currents would hasten anoxia and a winter fish kill condition. Density currents are no doubt greatly affected by weather conditions from year to year perhaps explaining the sporadic nature of winter fish kills. The fish kills may also be the result of extremely cold periods which result in thicker ice and a reduced lake water volume. This would increase the sediment surface area:volume ratio and aggravate or worsen the oxygen depletion rate.

The possibility exists that fish, particularly the large-mouthed bass which is better adapted to sloughs and mud-bottom lakes, can survive the very low oxygen levels in the lake by congregating near small inflow streams or in areas of groundwater inflow where

oxygen levels are somewhat higher. These sources of oxygen may be available throughout the winter except again during extremely cold years.

Understanding the mechanisms and predicting the oxygen depletion in Tie Lake would require considerably more study. Indeed, the prediction of winter fish kills and sediment nutrient release resulting from oxygen depletion may not be possible. However, it is safe to conclude that Tie Lake, because of its high sediment surface area:lake volume ratio (low mean depth) will always suffer from severe oxygen depletion under ice-cover regardless of reduced productivity or minor changes to lake volume.

4.2 Trophic Status and Calcite Precipitation

The blue-green algal bloom, first observed in the fall of 1980, was characterized by a pea-green colour and greatly reduced water clarity (Secchi depth about 1 meter, Figure 6). Examination showed high levels of chlorophyll *a* and total phosphorus (Figure 5) classifying the lake as meso-eutrophic. This condition persisted throughout the period of ice-cover, from mid-November to the end of March, and through the early spring. Lake samples on April 7 and May 5 recorded similar levels of productivity. On May 14 the author found the appearance of the lake unchanged from May 5. By May 28 however, the lake had cleared and the opaque green colour of the phytoplankton bloom was gone. The June 10 lake sample showed greatly increased water clarity

(Figure 6) and a dramatic drop in total phosphorus and chlorophyll a levels (Figure 5). In addition the phytoplankton population had greatly changed in composition from being dominated almost entirely by the blue-green Microcystis to a variety of greens and flagellates (Table 7). As previously suggested, the reason for this dramatic reduction in algal productivity is thought to be the coprecipitation of phosphorus with calcium carbonate.

Evidence of calcite precipitation is not conclusive due to the unfortunate curtailing of analysis for several key parameters, namely dissolved calcium and magnesium and total alkalinity (Table 1). A significant drop in specific conductance on June 10 (Table 1) indicates the removal of a significant quantity of dissolved ions from the water column. The calcite precipitation recorded in the lake in 1973 by Buchanan and Goddard showed a reduction in calcium from 30-40 mg/L to 19 mg/L. Winter concentrations of calcium in 1980-81 were similarly in the order of 40 mg/L (Table 1). In 1973, post-precipitation magnesium levels were about 26 mg/L compared to winter 1980-81 levels of about 28 mg/L which is consistent with the very slight drop in dissolved magnesium expected after carbonate precipitation, typically <99% being in the form of $CaCO_3$ (Otsuki and Wetzel 1972). Post-precipitation hardness levels in 1973 were about 155 mg/L compared to pre-precipitation levels in 1980-81 of about 210 mg/L which is again consistent with a reduction in

dissolved calcium due to precipitation. Similarly, pre-precipitation total alkalinity levels in 1973 were about 224 mg/L compared to 1980-81 levels of about 230 mg/L while post-precipitation levels in 1973 were reduced to 168 mg/L.

Perhaps the greatest support for calcite precipitation and phosphorus coprecipitation in Tie Lake in 1981 is the rapidity with which total phosphorus and chlorophyll a levels dropped, more dramatically observed as a clearing of the water column which occurred within 14 days. Initially, the cause was suspected as being the flushing of a large portion of phosphorus from the lake as a result of heavy rainfall. On May 5 the average total phosphorus concentration in the lake was 30 µg P/L which represents 72 kg of phosphorus in the total lake volume. The average loading of total phosphorus leaving the lake via the outflow stream during this period was estimated at 105 g/d or a total of 4 kg of phosphorus for the 37 days between sampling from May 5 to June 10. Assuming no inputs of phosphorus during this period, which of course was not the case, this would leave 68 kg P in the lake giving an average lake concentration of 28 µg P/L, considerably greater than the 14 µg P/L measured on June 10.

It is interesting that in 1973 the levels of total phosphorus were not greatly affected by calcite precipitation being about 15 µg P/L before and 12 µg P/L after and these concentrations were similar to post-precipitation levels in 1981 (13-14 µg/P/L).

This suggests that there may be a fraction of the total phosphorus that is unaffected by calcite precipitation. Unfortunately no measure of biomass was done in 1973 for compariosn. Total phosphorus levels prior to precipitation in 1973 were one half to one third of those in 1980-81 and presumably biomass was similarly lower. Certainly the Secchi depths throughout 1972 - 73 indicated clear water (3 meters or greater). The reasons for the great increase in lake phosphorus concentration over the seven years are discussed in the next section.

Recent research has found that coprecipitation of phosphorus with calcium carbonate is a common occurrence in temperate, hardwater lakes and is often induced by photosynthetic activity of planktonic, benthic and littoral algae (Otsuki and Wetzel 1972; Wetzel 1975; Murphy et al 1983). The photosynthetic activity of the alga produces a microenvironment around the cell which may easily have a pH of 9 (Otsuki and Wetzel 1972) and thus act as nucleation sites for $CaCO_3$ precipitation reactions which, once started, proceed rapidly providing the water is saturated with respect to $CaCO_3$ (Murphy et al 1983). The $CaCO_3$ precipitate may actually adhere to the algal cells much as it does on macroalgae like Chara which feel crusty and rough from calcite deposites (McKean, 1984, pers. comm). The phytoplankton cells are thus settled explaining the extremely rapid clearing of the water column. Calcium carbonate precipitation may occur abiotically

precipitation may occur abiotically forming fine particles or colloidals though at a much slower rate than metabolically induced precipitation (Wetzel 1975).

Otsuki and Wetzel (1972) found that below pH 9 the coprecipitation of phosphorus increased with increasing temperature because of a proportional reduction in $CaCO_3$ solubility and that this relationship was more pronounced at higher $CaCO_3$ concentrations. They also found that the reaction may take place extremely fast (10 minutes for 75% PO_4 precipitation at pH 9.5 and 24° C). On May 5 the average lake temperature was about 11° C and on June 10 about 16° C.

It appears then that the coprecipitation of phosphorus induced by algal photosynthesis is an indirect density dependent population control (Otsuki and Wetzel 1972). Murphy et al (1983) found this to be the case in Black Lake, a small naturally hypertrophic lake in the southern Okanagan, enriched by nearby phosphate rock deposits and experiencing seasonal blooms of the blue-green Aphanizomenon. In deeper, seasonally stratified lakes such as Black Lake, phosphorus coprecipitation occurs only in the epilimnion so that spring and fall overturn or thermocline breakdown from strong winds can replenish the phosphorus from the hypolimnion (Otsuki and Wetzel 1972; Murphy et al 1983). In Tie Lake stratification does not occur so that the entire lake is subject to phosphorus coprecipitation, sediment

desorption and external inputs then are the only means available to build up phosphorus concentration. This probably means that the cycle of phosphorus buildup, increased algal production and photosynthetically induced phosphorus coprecipitation is less predictable than in a deeper, dimictic lake if in fact it is predictable at all. In the absence of stratification and twice annual mixing the phosphorus accumulation in Tie Lake is dependent wholly on input and output. The inputs of phosphorus to the lake, both natural and man-made, are highly variable, dependent on levels of human activity, weather patterns and variations in the annual hydrologic cycle, which also affects the output.

Calcite precipitation in Tie Lake is as much a function of $CaCO_3$ saturation as it is the level of photosynthetic activity. Indeed, the precipitation event recorded in 1973, when total phosphorus levels and presumably algal productivity were much lower than in 1981, suggests that large algal blooms are not necessary to induce calcite formation. Once the lake is supersaturated with respect to $CaCO_3$, much lower levels of algal productivity may induce precipitation. In any event, the prediction of the frequency of calcite precipitation events in Tie Lake and a determination of the conditions under which they could be expected will require considerably more study. This natural phenomenon is never the less of great importance in the maintenance of acceptable lake water quality.

4.3 Phosphorus Loading and Water Quality

Having established that phosphorus is the factor limiting phytoplankton growth, the available supply of this nutrient can be estimated. Much research over recent years on a great many different lakes has led to the development of mathematical relationships or empirical models relating phosphorus concentration, loading, lake morphometry and lake hydrology to each other. (Vollenweider and Dillon 1974; Dillon and Rigler 1975, Larsen and Mercier 1976; Reckow and Simpson 1980). These models, often using an minimum amount of field data, are used to predict changes in lake water quality when individual parameters, such as phosphorus loading, are altered. Most of the lakes used in the development of these mathematical models are lakes exhibiting a well developed summer stratification. Such lakes produce much higher correlations between model parameters and on the whole behave in a more predictable manner than do shallow, eutrophic lakes (Larsen and Mercier 1976; Reckow and Simpson 1980).

Tie Lake is obviously not well suited to the use of such empirical models. Despite the large inherent error expected, a model developed by Dillon and Rigler (1975) has been used to illustrate the amount of change that could be expected in spring overturn phosphorus concentration with changes to the flushing rate and phosphorus loading rate. The equation is written as:

$$[P] = \frac{L(1-R)}{Zp} \qquad \text{equation 1}$$

where [P] = the spring overturn phosphorus concentration in mg/m^3, L = the phosphorus loading in $gP/m^2/yr$, R = the phosphorus retention coefficient, Z = the mean depth in meters and p = the flushing rate in years^{-1}. Spring overturn phosphorus concentration was measured directly in 1981. Phosphorus loading and flushing rate were estimated and will be discussed in some detail. The mean depth was determined from surveys done in the past (Buchanan and Goddard, 1976). The phosphorus retention coefficient is essentially a representation of the difference between the amount of phosphorus entering the lake from various sources and that leaving the lake via the outflow stream. This parameter is one of the most important in the equation and unfortunately is also the most difficult to measure or estimate. Another empirical model developed by Larsen and Mercier (1976) has been used to estimate the retention coefficient. The first equation used is written as:

$$R = 0.482 - 0.112 \ln p \qquad \text{equation 2}$$

where p = the flushing rate in years $^{-1}$. The second is written as:

$$R = 0.854 - 0.142 \ln qs \qquad \text{equation 3}$$

where qs = the areal water loading rate in m/yr. These two methods of estimating phosphorus retention coefficient are examined here to determine which is most realistic. The wide range of resulting values illustrates the great variability and uncertainty that can be expected when applying empirical models such as these to a lake which differs significantly from the lakes used to initially develop the models.

4.3.1 Water Balance

The water supply to the lake was calculated from inflow stream flow guaging done in 1981, precipitation data and correlations with continuous flow guaging from nearby streams. These calculations produced an estimated average annual runoff to the lake of 480 dam^3/yr(dam^3 cubic decameters = 0.81 acre feet). In 1980 the runoff was below average at 380 dam^3/yr. and in 1981 above average at 530 dam^3/yr. (Barr, 1982). It must be stressed that these figures are rough estimates only, more accurate estimates would require a long term study of lake level fluctuations and continuous flow guaging.

A comparison of annual runoff entering the lake with annual precipitation shows no direct relationship. The runoff entering the lake in 1980 was 21% below the average and yet the annual precipitation for that year, as measured at the Cranbrook airport, was 37% greater than the 1955-1980 average. In 1981 the runoff was 10% greater than average while precipitation was 36% greater. Empirical evidence has shown runoff of a non-guaged watershed can be better estimated by extrapolating from and is more strongly correlated to a nearby guaged watershed than to local precipitation, (Barr, 1983). The amount of runoff from a watershed is greater when the precipitation falls as rain rather than as snow (Barr, 1983). This may partly explain the greater runoff to Tie Lake in 1981 over 1980. Total precipitation in both years was about the same, however in 1981 79% fell as rain compared to 64% in 1980.

Historical information from measurements taken at Elko showed that evaporation was approximately equal to direct rainfall on the lake surface so that the net affect on the lake's water supply is considered to be zero (Barr, 1983).

An extremely important factor affecting the trophic status of any lake is the water residence time usually expressed as the number of years required to completely replace the lake volume. The flushing rate is merely the inverse of this or the portion of lake volume replaced in one year. The lake volume is simply calculated as the product of the lake surface area and the mean depth. Previous surveys determined the lake surface area to be 1.26 km^2 and the mean depth to be 2.19 m (Buchanan and Goddard, 1976). However, a complication arises with regard to the mean depth in that previous to the fall of 1980 the outflow from the lake was via a culvert through an earthen dam. During the fall of 1980 a channel was cut through the dam to an elevation some 40 cm below the culvert. Since the lake level seldom reaches the bottom of this channel it is estimated that the mean depth has been reduced by 30 cm to 1.9 m. The corresponding reduction in lake volume means a slight reduction in water residence time or a slightly increased flushing rate given the same water supply. Table 9 shows water residence times and flushing rates calculated for the various conditions that existed through the study period as well as for an average year.

TABLE 9 Calculation of water residence times and flushing rates from runoff estimates.

	lake volume (V) dam^3	water supply (Q) dam^3/yr	water residence time (tw=V/Q) yrs	flushing rate (1/tw) 1/yrs
average annual				
with dam	2760	480	5.7	0.17
without dam	2394	480	5.0	0.20
1980 (with dam)	2760	380	7.2	0.14
1981 (without dam)	2394	530	4.5	0.22

It can be seen that lowering the levels of the outflow alone does not significantly alter the flushing rate. Instead of replacing 17% of the lake volume in an average runoff year, 20% of the smaller volume would be replaced. In 1981 the smaller volume coupled with the increased water supply resulted in the replacement of an additional 8% of the lake volume over the 14% in 1980. The significance of these changes in flushing rate will become apparent later in the report.

Another method of calculating lake flushing rate is by direct measurement of the outflow volume. This has been estimated for Tie Lake from the seven instantaneous flow rates taken through the spring, summer and fall of 1981. The average flow between each gauging was used to determine the total volume of water leaving the lake for that period, these volumes were then summed to give an estimate for the year.

The flow rate for the winter period was estimated at a level somewhat lower than the rates guaged in the spring and fall which were quite similar. Though also only a very rough estimate, this produced a volume of approximately 1000 dam^3/year leaving the lake, practically twice that of the inflow estimate of 530 dam^3/year for 1981. The reasons for this wide discrepancy are not obvious, both estimates being based on very little data. The possibility exists that some of the groundwater that appears to enter the lake may emanate from outside the small surficial drainage basin. This groundwater could originate from the mountains to the north of the lake and flow south over the bedrock which underlays the Tie Lake region. The known "springs" entering the lake are indeed located on the north shore.

When applied to the same lake volume (ie: without the dam) this greater outflow volume produces a shorter residence time of 2.4 years and flushing rate of 0.42/yr. As before, increasing the lake volume by

raising the elevation of the outflow 30 cm. has little effect on the flushing rate which decreases by 6% to 0.36/yr. (residence time 2.7 years).

Flushing rate estimates calculated from the 1981 outflow are considered in this report to be closest to the true value since the measurement of a single pathway of outflow is likely to involve less error than estimations of the multiple pathways of inflow (inflow streams, sheet runoff, groundwater etc.). The true water residence time in all probability lies somewhere between the 2.4 and 4.5 years with 3.0 years being a reasonable estimate.

4.3.2. Phosphorus Sources

Phosphorus finds it way to a lake from a number of different sources through a number of different pathways. In addition to water born forms, phosphorus may enter a lake from the atmosphere as wet and dry fallout. In no particular order the various sources of phosphorus considered are as follows:

inflow streams - carry P from various sources
domestic waste - lakeside septic disposal systems
livestock waste - cattle ranging in the drainage basin
atmosphere - precipitation and dry fallout on the lake
internal loading - recycling P released from lake sediment

water fowl - large numbers use the lake each spring
lawn fertilizer - lakeside lawns
boating - disturbed sediment releasing dissolved phosphorus

Inflow streams

The five inflow streams (Figure 1) were sampled for phosphorus and flow guaged seven times through 1981. Instantaneous loadings of each stream were added to produce a total stream inflow loading. The average loading between each sample date was used to determine the mass loading for each period. The fall loading, about 1/10 of the spring values, was used to estimate the winter period during which no data was gathered. The total phosphorus load for 1981 was estimated to be 2.5 kg from all surface waterinflow.

The sources of phosphorus entering the lake via the inflow streams include livestock and domestic waste as well as some background or natural sources. The possibility of double-counting when estimating loadings from livestock and septic systems is likely, however, error involved is considered negligible due to the small loading from the inflow streams.

Domestic Waste

Phosphorus loading estimates from lakeside domestic waste disposal were done by Wiens (1983), assumptions and methodology used are explained in detail in that report.

The numbers and distribution of lakeside dwellings were obtained from various sources including the Tie Lake Ratepayer's Association, the Regional District of East Kootenay and the Assessment Authority of B.C.

Of the total loading of 44.6 kg/yr. from this source 27.8 kg/yr. was attributed to the 22 permanent homes and 16.8 kg/yr. to the 77 summer homes in the immediate vicinity of the lake. Permanent dwellings obviously contribute much more phosphorus on an annual basis due to the continuous discharge of waste compared to an assumed 2.5 month discharge for summer homes (Wiens 1983). At Tie Lake the permanent homes account for only about 22% of the total number of dwellings but contribute 62% of the phosphorus loading from this source. The greatest increases in phosphorus loading from domestic waste in the future will thus be a result of the number of additional permanent homes established, either through new development or through the conversion of summer residences.

Sewage disposal systems for summer homes are either a septic tank/tile field system or pit privy and greywater rock pit combination. The proportions of each for the 77 summer homes at Tie Lake was assumed to be roughly 50% (Pagarut 1982, pers. comm.). With chis in mind, 9.9 kg P/yr. or 59% of the 16.8 kg P/yr. was attributed to septic tanks and fields, 5.8 kg P/yr. or 34% was attributed to greywater rock pits

and only 1.0 kg P/yr. or 6% to pit privies. The use of large quantities of water for sewage disposal tends to increase the phosphorus loading by providing a means of transport. With identical potential loadings per capita the human waste portion of the estimated loading from septic tanks was about 9.9-5.8 = 4.1 kg P/yr or 24% of the total while pit privies contributed an estimated 1.0 kg P/yr. or 6%.

Livestock

Phosphorus loading estimates from cattle grazing within the Tie Lake Drainage basin were also done by Weins (1983). Information on the numbers and distribution of cattle and pasture use were obtained from the B.C. Ministry of Forests, Range Section, the agency responsible for management of crown range.

Each summer 350-425 head of cattle range on each of the three pastures illustrated on Figure 1 for approximately one month each (Gibbard, K, pers. comm 1982). Prior to 1980 the pastures were not fenced into three separate units and cattle ranged more freely. Approximately 75% of the Tie Lake pasture lies within the drainage basin and cattle have easy access to a large portion of the northwest lakeshore. Only a small part of the Relief Camp pasture lies within the drainage basin and cattle can access the lakeshore at the extreme west tip of the lake. (see Figure 1 and Weins, 1983). The Tie Lake seeding is an 80 hectare pasture lying entirely within the basin with its southern boundary located about 100 m. north

of the lake. In 1979 the area was cleared, fenced and about 75 ha was planted with alfalpha and crested wheat grass (Gibbard, K, pers. comm 1983). Fertilizer containing only nitrogen is applied once every 3 or 4 years. Of a total estimated phosphorus loading to the lake from cattle of 47.7 kg P/yr. the Tie Lake seeding pasture accounted for 36.2 kg P/yr. or about 76%. The Tie Lake pasture contributed 9.9 kg P/yr. or 21% and the Relief Camp pasture 1.4 kg P/yr. or 3%. The large disproportionate contribution of phosphorus from the Tie Lake seeding pasture was attributed to the high density of cattle, coarse soil texture and close proximity to the lake (Wiens 1983). Wiens (1983) estimated that approximately 92% of this 36.2 kg P/yr. from the Tie Lake seeding pasture reached the lake via sub-surface or groundwater pathways with only 8% being carried by surface streams.

Atmosphere

Atmospheric inputs of phosphorus to lakes are typically a signficant proportion of the total loading. The determination of this quantity for individual lakes is extremely difficult, rainfall and dustfall analyses usually produced greatly varying results requiring expensive long-term studies to produce better estimates. Several dustfall samples taken at Tie Lake showed this extreme variation in phosphorus content and estimates from in-depth research in the literature were considered more reliable.

A literature survey of contemporary research has produced an estimated atmospheric phosphorus loading to Tie Lake of 20 kg/km^2/yr (Gilliom 1981; McKean 1982; EPA 1980). Atmospheric phosphorus loading can be very high if a lake is located near industrial point source discharges such as pulpmills, mines or factories. No such discharges are found in the vicinity of Tie Lake.

This estimated loading from the atmosphere of 20 kg/km^2/yr. applied to the lake surface area produces a mass loading of about 25 kg/yr.

Internal Loading

This source is essentially defined as the release of phosphorus from the lake sediments during periods when oxygen depletion of bottom water produces an anaerobic or reducing condition at the sediment-water interface. This normally takes place during summer thermal stratification or under ice cover. The presence of phosphorus sediment release can be detected through increased levels of dissolved phosphorus in bottom water samples during such periods. Though data is not plentiful, samples on November 20, 1980 and February 25, 1981 do not indicate phosphorus release from sediments (Table 5). As such the phosphorus loading from this source is considered non-existent or at least negligible.

Water Fowl

During the spring large number of water fowl were observed on the lake engaged in nesting activities. The excrement from these birds no doubt contributes some phosphorus to the lake during this period, however, since April and May samples showed no increases in phosphorus it is felt that this source was negligible.

Lawn Fertilizer

The use of chemical fertilizers containing phosphorus on lakeside lawns no doubt contributes some phosphorus to the lake. The estimation of the loading from this source is difficult and highly variable depending on the type of fertilizer, application rate, soil type, rainfall and irrigation practices. Because of these difficulties no estimate from this source has been done although it is not expected to be significant. Although accurate quantification is complicated and time consuming, the use of lawn fertilizers containing no phosphorus can completely eliminate this source.

Boating

The concern here is that outboard motors may disturb lake sediment in shallow littoral areas which may in turn release dissolved phosphorus into the water column. Despite the fact that at times considerable

sediment is stirred up by water ski activity on Tie Lake this source is considered negligible primarily because of the lack of sediment phosphorus release (internal loading). A study by Butcher (1982) also found that outboard motor use is seldom related to lake eutrophication in this way.

Total Phosphorus Loading

The estimated total annual mass loading of phosphorus to Tie Lake is summarized as follows in Table 10:

TABLE 10 Estimated Mass Phosphorus Loading Entering Tie Lake

	kg P/yr	% of total
septic system		
22 permanent homes	28	23
77 summer homes	17	14
Sub-Total	45	37
livestock		
Tie Lake seeding	36	30
Tie Lake pasture	10	8
Relief Camp pasture	2	2
Sub-Total	48	40
atmosphere	25	21
inflow streams (background)	2	1.5
TOTAL	120	100

The most significant conclusion here is that 77% of the total phosphorus load to the lake originates from human activity, about equally divided between lakeside domestic waste disposal and cattle ranging in the watershed. Of the 48 kg P/yr arising from livestock 36 kg P/yr. originates from the Tie Lake seeding pasture which amounts to about 30% of the total phosphorus loading from all sources.

It is suggested that the concentration of the entire range herd in the Tie Lake seeding pasture, on coarse textured soils and very close to the lake, has significantly increasad the total phosphorus loading from livestock ranging throughout the drainage basin. This increased loading to the lake has taken place without any increase in the number of cattle but merely by changing the way in which the range is utilized.

4.3.3 Predictive Modeling

As previously mentioned the relationship developed by Dillon and Rigler (1975)(equation 1) was used to predict changes in lake phosphorus concentration with two estimates of phosphorus retention coefficient developed by Larsen and Mercier (1976) (equations 2 & 3). The first step in this approach was to test the model against the actual estimated phosphorus loading using the spring overturn phosphorus concentration monitored in 1981. Next, the affect

on lake phosphorus concentration from an increased flushing rate was investigated and finally the affect of reducing or increasing the phosphorus loading rate was examined.

Model Test Using 1981 Data

In theory, working back from an actual spring overturn phosphorus concentration, the calculated phosphorus loading should be similar to the value estimated from the various sources. To this end, equation 1 can be rewritten:

$$L = \frac{\bar{Z}p.[P]}{1-R} \qquad \text{equation 4}$$

For Tie Lake in 1981 the mean depth ($\bar{Z}$)= 2 meters, the spring overturn phosphorus concentration ([P]) = 30 mg/m^3 and the two estimates of the flushing rate (p) were: from inflow estimates p = 0.22/yr. and from outflow estimates p = 0.42/yr. Two methods of determining phosphorus retention coefficient (R) were examined, one using areal water loading (equation 2) and one using flushing rate (equation 3)(Larsen and Mercier 1976). The areal loading (q_s) is simply calculated by multiplying the flushing rate (P) by the mean depth ($\bar{Z}$). Using the inflow estimates q_s = 0.42 m/yr and using outflow estimates q_s = 0.8 m/yr. Thus using equation 2:

$$R = 0.854 - 0.142 \ln q_s$$

Gives us R = 0.98 for the inflow estimates and R = 0.88 for the outflow estimates. Using equation 3:

$R = 0.482 - 0.112 \ln p$

R = 0.65 for the inflow estimates (p = 0.22/yr) and R = 0.58 for the outflow estimates (p - 0.42/yr). Empirically, the estimates of R from equation 3 are considered the most realistic. Typically, phosphorus retention coefficients approaching 1.0, such as those arrived at using equation 2, are characteristic of large lakes with water residence time in excess of 50 years (Nordin 1983, pers. comm.). It is important to note here that a doubling of flushing rate when using equation 3, from 0.22/yr. to 0.42/yr., does not drastically alter the phosphorus retention coefficient, R = 0.65 and R = 0.58 respectively. The constants in the formula, calculated from a number of test lakes are far more important which is why a large potential error is introduced when the lake studied is dissimilar to the lakes used to develop the model. This error is then transferred to the Dillon and Rigler model (equations 1 and 4) in which R is an extremely important factor.

Using equation 4, when inflow estimates are used to calculate p and thus R, the areal phosphorus loading is 36 $mg/m^2/yr$ or a mass loading of 45 kg/yr. When outflow estimates are used a value of 60 $mg/m^2/yr$ or 76 kg/yr. are produced. Both estimates are somewhat below the 120 kg/yr derived from source estimates. In all likelihood this is the result of inherent model error already discussed. Despite this poor model fit equation 2 can be used to examine the degree of change to the lake phosphorus concentration, that can be expected when the flushing rate is increased or when

phosphorus loading is reduced. Unfortunately however the actual calculated phosphorus concentrations themselves will not be reliable.

Consequences of Increased Flushing Rate

The effect of raising and lowering the elevation of the outflow channel on the flushing rate of the lake, as previously mentioned, is minimal. The lowering of the channel in 1981 by 30-40 cm. only increased the flushing rate an estimated 3-6% while producing undesirably low lake levels in the fall and winter. In this case the outflow channel was lowered for the entire year, the installation of a control dam to allow more water to escape only during high water would affect the flushing rate even less. The intended purpose of increasing the flushing rate is to reduce the lake phosphorus concentration. The small decrease in concentration due to the slight increase in flushing rate resulting from lowering the outflow channel would likely be offset by the reduced lake volume which would tend to increase the phosphorus concentration.

In order to significantly increase the flushing rate, the water supply to the lake must be increased. A diversion of a portion of the flow of nearby Little Sand Creek is the most obvious means of accomplishing this. Considering the logistics and available water, a diversion flow of about 1.0 CFS (0.028 m^3/sec) is considered feasible (Peters 1983, pers. comm.). Table 11 illustrates the effect that a 1CFS flow over varying periods would have on the flushing rate and

considered feasible (Peters 1983, pers. comm.). Table 11 illustrates the effect that a 1CFS flow over varying periods would have on the flushing rate and thus the spring overturn phosphorus concentration. These estimates are based on the 1981 lake volume when the outflow channel was lowered. Were the channel to be raised because lake levels are too low the flushing rate would be somewhat lower starting at 0.36/yr. The proportionate effects of the various diversion schemes would be the same.

The actual quantity of water that can be diverted will be subject to various restrictions. These include insuring that minimum flow requirements for fish in Little Sand Creek are met and giving user priority to existing downstream water licenses.

These estimates of the reduction in lake phosphorus concentration through increased flushing rate assume no increase in phosphorus loading from the diversion. Phosphorus levels in Little Sand Creek are probably quite low however if diverted water is allowed to flow to the lake via natural drainage channels it may pick up additional phosphorus from livestock sources. This would affect somewhat the benefits of increased flushing rate although the effect is not expected to be great.

TABLE 11 Changes in the Flushing Rate and Spring Phosphorus Concentration Expected from a 1.0 CFS Diversion of Little Sand Creek

Duration of Flow	Volume Added (dam^3)	Flushing[1] Rate (yr^{-1})	Spring [P][2] ($\mu g/L$)	Percent Decrease in[P]
no diversion	-	0.42	30	-
2 months	147	0.48	27.2	9
4 months	293	0.54	25	17
6 months	440	0.60	23	23
12 months (continuous)	893	0.79	18.7	38

1.outflow estimates used to determine flushing rates
2.calculated using equation 1 L = 60 mg/m /yr. $\bar{Z}$ =Zm calculated using equation 2 = 0.58 (initially).

The point of entry to the lake of any diversion and its proximity to the outflow channel will also be a factor in determining the benefits of an increased flushing rate. If the diversion enters the east basin the flow will likely short circuit to the outflow and have less effect on the west and central basins.

Table 11 shows that at least six months of a 1CFS diversion would be required before any substantial drop in lake phosphorus concentration could be expected. Even with a continuous diversion at this flow rate the reduction does not equal that which occurred as a result of coprecipitation with calcite in 1981.

Consequences of Increasing or Decreasing Phosphorus Loading.

A reduction in the phosphorus loading to a lake is one of the most effective ways of reducing eutrophication. In addition it is often one of the most technically feasible and cost efficient solutions. The reduction of phosphorus loading from various sources is discussed and the effect on lake phosphorus concentration is estimated using the Dillon and Rigler (1975) model. The details of any single management strategy chosen, including the combined effect of also increasing flushing rate, will have to be calculated on individual basis since the number of options is infinite.

Prior to examining methods of reducing phosphorus loading the potential for future increases must be considered. Atmospheric sources are assumed to be steady state unless industrial activity in the vicinity increases. According to the Ministry of Forests the range use in the basin is at a maximum and should not change. This is presuming that another more intensively managed pasture such as the Tie Lake seeding is not developed within the watershed. Residential development around the lake, however, is not at a maximum. Additional phosphorus loading can be expected from the future development of approximately 30 additional unimproved lots and from the conversion of a number of existing summer homes to permanent residences.

The assumption is made that approximately half of the existing summer homes could ultimately become permanent. In order to estimate the increase in phosphorus loading from this greater annual population a simple loading per house for each category was calculated from the existing numbers and relative loadings. This method does not take into account factors such as different soil types and distances from the lake used in initial estimates done by Weins (1983) and as such can only be considered as very rough estimates. To include these factors and make a detailed calculation would involve predicting how many homes in each sub-area (see Weins 1983) would become permanent. This would be very difficult if not impossible and as such the more generalized calculation has been used.

From Table 10:

a) the estimated phosphorus loading per dwelling is:
   summer: 17 kg/yr - 77 homes = 0.22 kg/house/yr.
   permanent: 28 kg/yr - 22 homes = 1.3 kg/house/yr.

b) conversion of half the summer dwellings to permanent would adjust the total loading from existing residential development as follows:
   summer: 38 homes x 0.22 kg/house/yr = 8.4 kg/yr.
   17 - 8.4 = 8.6 kg/yr
   permanent: 38 homes x 1.3 kg/house/yr=49.4 kg/yr.
   28 + 49.4 = 77.4 kg/yr.
   TOTAL 8.6 + 77.4 = 86 kg/yr.

An additional loading from new development on as yet unimproved lots has been calculated as follows:

a) only the 12 undeveloped lakeside lots are considered since loading decreases greatly with distance from the lake (Weins 1983).
b) of the 12 new residences it is assumed that 6 would be permanent and 6 would be summer use only.
c) permanent: 6 x 1.3 = 7.8 kg/yr.
summer: 6 x 0.22 = 1.3 kg/yr.
TOTAL 7.8 + 1.3 = 9 kg/yr.

This brings the total phosphorus loading from all residential development, both existing and potential, to 86 + 9 = 95 kg/yr. This represents a 110% increase over the present estimated loading with the dominant factor being the number of additional permanent residents.

This rough estimate of the potential increase in phosphorus loading from domestic waste sources brings the overall loading to the lake to about 170 kg P/yr, an increase of about 41%. Using the Dillon and Rigler model (1975) (equation 1) this would also produce a 40% increase in lake phosphorus concentration from 30 µg/L to 42 µg/L. For reasons already explained, this calculation was done using a percentage increase in the areal P loading (60 $mg/m^2/yr$ + 42% = 85 $mg/m^2/yr$) rather than the actual loading estimate. In addition, these estimates were made using the present flushing rate of 0.42/yr. An increased flushing rate would obviously mitigate somewhat the effects of increased P loading from greater lakeside development. For example, a 6 month diversion at ICFS, which results in a flushing rate of 0.6/yr (Table 10) would produce a lake phosphorus

concentration of 32 µg/L or a 6% increase with the maximum predicted level of development rather than the 42 µg/L or 40% increase expected at the present flushing rate.

Using the Dillon and Rigler (1975) model it can be shown, as in the above example that changes in phosphorus loadings entering the lake are directly proportional to spring lake phosphorus concentrations. In other words a 30% reduction in phosphorus loading, for example, would reduce the spring overturn phosphorus concentration by 30% from 30 µgP/L to 21 µgP/L. It remains then to determine the best ways to reduce the phosphorus loadings. The targets for this effort should obviously be the most significant yet manageable sources, namely livestock and domestic waste.

5. CONCLUSIONS

a. The total concentration in the lake in 1980-81 was two to three times the levels recorded in 1972-73. This is thought to be the result of increased quantities of domestic waste from growing lakeside development and increased nutrient inputs from livestock ranging in the basin.

b. The increased phosphorus concentration in the lake (30 µgP/L at spring overturn 1981) is thought to have caused the nuisance bloom of blue-green algae in the late summer of 1980 which persisted until May 1981.

c. High levels of dissolved calcium in the lake in conjunction with photosynthetic consumption of carbon dioxide during the bloom are thought to have resulted in the rapid precipitation of $CaCO_3$ in the latter half of May 1981. The calcite in turn coprecipitated sizeable quantities of phosphorus lowering the lake concentration by 50%. At the same time the phytoplankton bloom disappeared either due to a settling of cells encrusted with calcite deposits or a large die-off caused by phosphorus starvation.

d. At the present or increased phosphorus loading the lake phosphorus concentration will eventually reach high enough levels to cause another algal bloom similar to that of 1980. The calcite/phosphorus coprecipitation phenomenon is thought to be cyclic, and how this will affect the accumulation of phosphorus in the lake is unknown. From time to time the coprecipitation of phosphorus temporarily lowers the lake phosphorus concentration before it reaches problem levels thereby delaying the inevitable return of a large algal bloom.

e. The depletion of oxygen under ice, which causes periodic fish kills, is primarily a function of the large sediment surface area: lake volume ratio. The eutrophication of the lake, caused by human activities, will aggrevate this condition, increasing the oxygen depletion rate over a number of years.

6. POSSIBLE METHODS FOR IMPROVING WATER QUALITY

Several measures should be considered if eutrophication of Tie Lake is to be reduced. Although these measures are

presented here separately, any combination of remedial measures will have to be considered on an individual basis.

6.1. Reduction of Phosphorus Loading

As the primary cause of eutrophication this measure is obviously one of the best permanent solutions. It can be broken down into the two major sources:

6.1.1 Domestic Waste

a) Removal of all waste from the basin through the construction of a collection system and treatment plant with discharge to infiltration lagoons located outside of the drainage basin. This approach would obviously be very costly, the use of holding tanks and removal to another treatment or disposal facility may be somewhat less expensive.

b) Improved operation and upgrading of individual disposal systems. This could include an inspection of private septic disposal systems to insure there is no short circuiting to the lake and the relocation of systems located close to the lakeshore to provide the maximum buffer zone available. This measure cannot be expected to reduce the overall phosphorus loading as estimated but will minimize additional quantities from poorly operating systems.

c) Restrict future development to prevent an increased phosphorus loading from domestic waste. A restriction on the number of permanent residents would be most effective.

6.1.2 Livestock Waste

a) Curtail the use of the Tie Lake seeding pasture. This would eliminate approximately 30% of the total annual loading and reduce the lake phosphorus concentration by a similar amount. The pasture could be relocated in another area farther from the lake or even out of the drainage basin.

b) Fencing of the northwest lakeshore within the Tie Lake pasture to provide a buffer zone from cattle. This would be quite expensive and since only 8% of the total annual phosphorus loading arises from this pasture would probably not be worthwhile.

6.2 Increasing the Lake's Flushing Rate

a) The use of a control dam on the outlet stream to lower lake levels and allow more water to pass through the lake. Changes made to the outlet in 1980 and 1981 lowered the outflow channel about 30-40 cm. and increased the flushing rate between 3 and 6% while at the same time producing undesirably low lake levels. The outlet elevation in this case was permanently lowered, the use of a control dam to allow more water out during high water only would

have even less effect on the annual flushing rate. The benefits of increasing the flushing rate toward reducing the lake phosphorus concentration are somewhat offset by the fact that the lake volume is also reduced. Because of the problems created by fluctuating lake levels for lakeside residents and the minimal effect on lake phosphorus concentration the use of a control dam is not recommended.

b) Diversion of water from Little Sand Creek. Increasing the water supply to the lake will indeed increase the flushing rate and decrease the lake phosphorus concentration. A 1.0 CFS flow is considered feasible which if continued for six months for example would increase the flushing rate by about 22%, based on the outflow channel elevation of 1981, and would decrease the average lake phosphorus concentration by an estimated 23%.

6.3 Lime Addition

The addition of quicklime (calcium oxide) to initiate the precipitation of calcium carbonate may be an effective, inexpensive treatment that could be used to reduce high levels of phosphorus in the lake. Calcium oxide is readily soluble and would raise the dissolved calcium concentration to saturation as well as raising the lake pH causing calcite precipitation to occur before it would naturally (McKean, 1984, pers. comm). Sampling the lake at spring overturn for phosphorus and calcium would indicate the need for such a treatment and give some idea of the dosage required.

It is suggested that the first treatment be done during a bloom to examine the effectiveness of this method and to gain information on dosage rates. Subsequent treatments could be done in early summer to lower phosphorus levels and prevent a large plankton bloom from occurring.

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POOR

QUALITY

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